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LCA FOR ENERGY SYSTEMS AND FOOD PRODUCTS

Inclusion of discard assessment indicators in fisheries life cycle assessment studies. Expanding the use of fishery-specific impact categories

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Abstract
Purpose The main purpose of this article is to propose specific discard indexes for their development in fisheries life cycle assessment (LCA). The objective of these is to characterize and standardize discards in worldwide fisheries.
Methods The global discard index (GDI) is intended to be an easily understood index whose use is extendible to any fishery in the world. It is presented as a dynamic index that aims to characterize and standardize discard rates between fisheries by direct comparison with the global discard rates reported periodically by FAO. Furthermore, a simplified approach excluding characterization is presented for scenarios in which the data quality linked to discards is poor. Two additional indicators, survival rate of discards and slipping, are proposed to improve the reporting and quantification of biomass waste by fishing vessels.
Results GDI implementation, together with two other fishery-specific impact categories, showed remarkable differences in the environmental impacts of several fishing fleets when compared with the obtained results for conventional impact categories. Results for the conventional categories were strongly influenced by the energy use in the fishery, while results obtained for fishery-specific categories presented variable trends due to the dependence on a wider range of factors. GDI inclusion favored direct comparison with worldwide average discard rates on a time scale basis, from a wet weight or a net primary productivity perspective, depending on the selected approach.
Conclusions Proposed indicators achieved the important objective of integrating discard data as a fishery-specific impact in fishery LCAs, increasing the benefits of implementing LCA in fisheries assessment. Specific advantages of these indicators include assessing changes in capture and landing composition, evaluating the selectivity of the fishing gears, and monitoring the behavior of fisheries in a normalized context respect to other fisheries. GDI was identified as an adequate methodological improvement for regular use in fisheries LCA. Future developments GDI include its harmonization for inclusion in damage assessment.

Keywords Discards · Fishery · Global discard index · Impact categories · LCA

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1 Introduction

Worldwide fisheries are subject to an outstanding stress due to overexploitation of stocks (Pauly et al. 2002; Christensen et al. 2003; Myers and Worm 2005). In fact, marine research centers forecast an increase of the stress on marine resources that anthropogenic activities are causing, since human population is growing rapidly and, thus, the need for extra protein sources (Tacon and Metian 2009). This has taken fishing stocks to their limit, with 84% of world fisheries being either overexploited or fully exploited (SOFIA 2010).

Growing seafood demand has been partially satisfied by a strong development of aquaculture. Unfortunately, the facilities that are developing fastest are those strongly dependent on wild caught fish for feed production (Naylor et al. 2000; Ziegler et al. 2003). Therefore, environmental assessment

schemes arise as an important mechanism to monitor the state of the art of world fisheries and aquaculture in order to provide guidance towards a scenario in which marine ecosystems are preserved, while assuring seafood supply to the world's population (Tidwell and Allan 2001; Worm et al. 2005).

Life Cycle Assessment (LCA), a standardized method for quantifying the environmental impacts associated with the life cycle of products (ISO 2006a, b), has shown to be a robust methodology for the environmental assessment of seafood. Nevertheless, it is generally accepted that LCA practitioners need to introduce a series of methodological innovations to increase the range of impact categories assessed (Pelletier et al. 2007). Most of these contributions should focus on the direct impacts generated on targeted stocks and their ecological implications (Reap et al. 2008; Buchspies et al. 2011), such as biodiversity loss, biotic resource depletion, and habitat alteration in marine ecosystems. However, such impacts are generally complex to quantify due to the lack of standard indicators and the variable effects that they can manifest in different locations (Ford et al. 2012). Furthermore, Pelletier et al. (2007) underline the importance of simplifying LCA results for these potentially new categories, with the main objective of broadening the target audience of the studies.

Consequently, seafood LCA studies in recent years have included biological concerns in the form of new impact categories, which include: seabed disturbance (Thrane 2006; Nilsson and Ziegler 2007; Ziegler et al. 2003; Ziegler et al. 2009); biotic resource use (BRU), first included in aquaculture studies as net primary productivity, NPP (Aubin et al. 2009; Papatryphon et al. 2004; Pelletier et al. 2009), but also implemented by Parker (2011); by-catch of non-target organisms (Ziegler et al. 2003, 2009); assessment of prematurely caught organisms (Emanuelsson 2008; Ziegler et al. 2009); and discard quantification (Vázquez-Rowe et al. 2010a, 2011a; Ziegler et al. 2011). All these newborn impact categories are yet to be standardized.

This situation shows that seafood LCA is slowly shifting to a more comprehensive framework for the environmental analysis of fisheries in terms of impact categories, in order to provide stakeholders with a more robust assessment to help them make choices (Ford et al. 2012). Yet some of these categories have been limited to reporting inventory data per functional unit (FU), which hinders the comparability between regions and processes (Milà i Canals et al. 2007). This is the case when reporting discard data. To our knowledge, LCA reports including these data have referred to this impact by accounting the total discard per FU. While discard quantification in fisheries LCA is a positive milestone, a current challenge is to deepen in the specific environmental impacts that discards may generate once discharged from fishing vessels.

Therefore, the ultimate goal of this article focuses on the proposal of a new set of potential indexes for use in fisheries LCA, combining the use of midpoint and endpoint level indicators (Bare et al. 2000). Specifically, the global discard index (GDI) is presented as an indicator that attempts to characterize and standardize discards in worldwide fisheries. To achieve this objective, two different approaches are suggested depending on data availability and quality. Additionally, an environmental assessment of a selected group of fishing fleets is developed including GDI and other fishery-specific impact categories.

2 Framework

2.1 Marine discards: an unresolved environmental problem in world fisheries

Discards, defined as the portion of the total organic matter of animal origin in the catch, dead or alive, which is thrown away, or dumped at sea for diverse reasons (Alverson et al. 1994), are an increasing matter of concern within the scientific community, given the enormous amounts of fish and other marine organisms that are returned to the ocean dead or damaged, and, therefore, altering the ecosystem (Stephen and Harris 2010). According to the latest report published by FAO (Kelleher 2005), the global marine discard rate in 1992–2003 was 8.0% (i.e., 7.3 million tons per year). Motives for discards are multiple and may vary between fisheries. Most of them, detailed in Table 1, are linked to environmental factors, to the gear used by the vessel or to a set of fishermen behavioral patterns, which may be influenced by management and economic issues (Catchpole et al. 2011).

High mortality rates can be observed in discarded organisms, especially within fish species (Cappell 2001; Catchpole et al. 2006). In this context, the direct effect on the marine ecosystems originated by discard mortality has been analyzed by several studies (Afonso et al. 2011; Benoît et al. 2010; Lindeboom and de Grott 1999; Mesnil 1996). In most cases, discard mortality has proven to (a) reduce biodiversity in fisheries worldwide (Greenstreet et al. 1999), (b) produce considerable variations when analyzing the relative abundance of species (Jennings et al. 1999), and (c) modify interactions between species (Christensen et al. 2003). Additionally, fishing mortality of discards has proven to vary depending on the used gear, since they infer different grades of damage on catch (Lindeboom and de Grott 1999).

Despite the scientific community agreeing that removing great quantities of non-desired biomass from world fisheries is unnecessary and harmful, many consequences of discarding are still unknown (Cook 2001; European Commission 2004; Kelleher 2005). Nevertheless, current policies consider that the lack of knowledge relating to discard effects should not delay improvement actions to reduce the amount of biomass discarded every year (Anon 2002; Catchpole et

Table 1 Reasons for discarding in worldwide fisheries (adapted from Clucas 1996)

Motive	Explanation
Resource motives	
Incorrect species	Not a target species for the vessel
Size requirements[a]	Certain individuals may be discarded for multiple reasons
Sex	Gender may be relevant in processing and marketing
Damaged fish	Due to mis-handling, predation or gear
Incompatibility	Could damage other species on board
Poisonous species	Poisonous or inedible species
Species spoils fast	This could accelerate spoiling in other species.
Management motives	
Space limitations	Usually temporal and economic scaling gives way to selectivity
Fishing quotas	Discarding individuals above maximum quota
Prohibition	Illegal to land a certain species
Season limitations	Some species are not allowed to be landed in specific times of the year, due to spawning, etc.
Gear limitations	Some species can only be caught with specific fishing gears.
Fishing grounds	Existence of administrative or protected areas where caught fish cannot be landed
Economic motives	
High grading	Sometimes related to size; individuals with less chances of being placed in the market will be discarded.

[a]Size requirements, despite having strong marine resource implications, can also be due to management and economic motives

al. 2005; Catchpole and Gray 2010; European Commission 2010).

Despite the fact that discarded organisms essentially belong to the same natural resource as landed catch, discards constitute a direct waste disposed off in the fishery, while landed species are transformed into industrialized products. Therefore, independently of the catch and quota reductions that may apply in each fishery, it is desirable to reduce the amount of discards respect to the total amount of captured fish, preferably improving catch selectivity rather than increasing the optimization of potential discards (Cook 2001; Kelleher 2005; Catchpole and Gray 2010). Hence, the perspective taken in this article, in accordance with previous studies, is based on considering discards and landed fish as two separate outputs obtained from the environment (Thrane 2006; Vázquez-Rowe et al. 2011b; Ziegler et al. 2011).

2.2 Specific framework for fisheries in LCA

The life cycle inventory (LCI) in an LCA study is based on compiling all relevant inputs and outputs linked to a particular process or service (PRè-Product Ecology 2011). To date, discard data were considered difficult to retrieve, which adds to the lack of an established mechanism for their inclusion in LCA (Weidema and Wesnaes 1996; Reap et al. 2008).

Additionally, it must be noted that discards are not considered co-products when analyzing the environmental burdens linked to the products. Therefore, discards are not computed when allocating impacts to the different products, since they are immediately returned to sea once their inappropriateness for landing is identified. In other words, it would not be desirable to attribute a specific environmental impact for conventional impact categories to discarded fish, since it would only reduce the associated impacts to the landed species, providing a misleading environmental profile of the marketable products.

3 Proposed indicators

To date, discard calculation in seafood LCA studies was limited to reporting the amount of generated discards per FU (Vázquez-Rowe et al. 2010b; Ziegler et al. 2009). This article attempts to develop methodological advances on how LCA studies may integrate discard data as an index rather than a mere value accompanying the other impact categories.

A total of three indicators have been proposed as indexes for discarding in fisheries. In the first place, the GDI is presented as a dynamic midpoint indicator to understand the relevance of discarding in a particular fishery (Fig. 1). Secondly, the survival rate of discards, an endpoint indicator, is discussed with the aim of integrating this crucial factor. A third indicator is linked to slipping operations in many purse seining fisheries. Finally, discussion on the potential impact that marine discards have on seabird populations is provided. The selected case study, developed in Sections "Application of the proposed GDI indicator to selected fisheries" and "Conclusions", only includes GDI

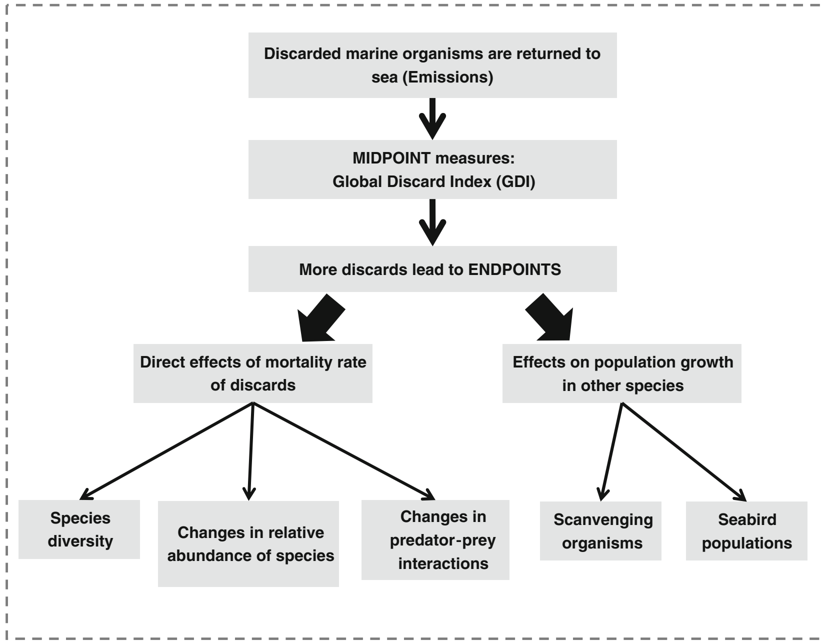

Fig. 1 Midpoint and endpoint modeling scenarios concerning discards in LCA

computation due to data availability limitations. Nevertheless, examples are provided for the other indicators when discussed.

3.1 Global discard index

3.1.1 Goal and scope

GDI is intended to be a straightforward indicator whose use is extendable to any fishery in the world to introduce discard quantification in fishery LCAs. A general flow diagram, including the relevant system boundaries, can be observed in Fig. 2. The methodology is based on the comparison of the discard rate for a certain fleet with the average worldwide discards considered as a reference value. For this, the latest available global discard rate reported by FAO is used as the reference set since it is considered the most accurate and current value (Kelleher 2005). Nevertheless, this value corresponds to data reported at least 8 years ago, showing that global discard rates are usually available with a significant time delay (Alverson et al. 1994; Kelleher 2005). Moreover, these average values are important reference points for fishery certification programs (Thrane et al. 2009). As summarized in Fig. 3, the proposed methodology comprises three major stages.

3.1.2 Required inventory data

The first step when using GDI in fishery LCAs is to obtain all the necessary data for LCI computation. Hence, apart from the regular inventory data required to carry out fisheries LCAs, it is important to include discards, catch rate, and the landing rate of the captured species, as detailed in Table 2. Discard reporting should follow the definition for this term given by FAO Report No. 547 (FAO 1996).

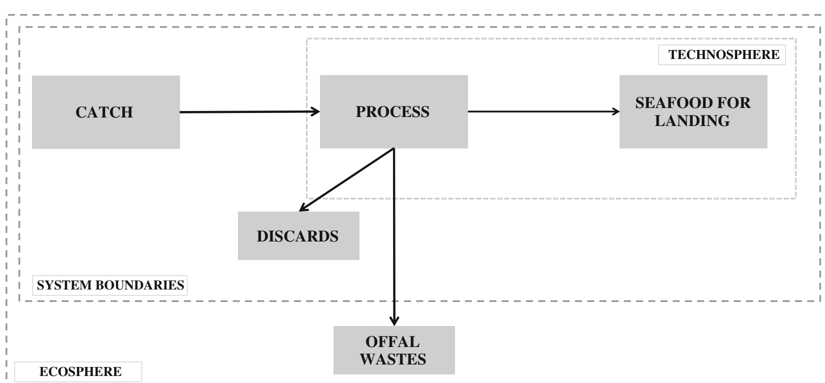

Fig. 2 Generic unit process for discards in fisheries and system boundaries

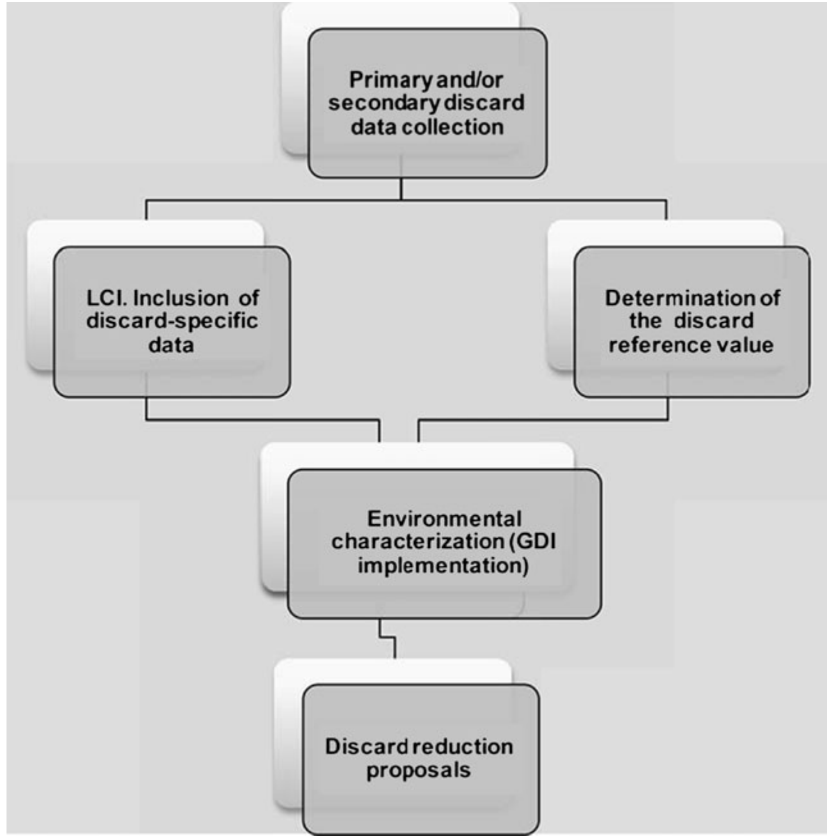

Fig. 3 Schematic representation of GDI implementation in fishery LCA methodology

3.1.3 Discard reference value and approach selection

Once the LCI is complete, the reference value that will be used when implementing GDI is determined. As mentioned above, the recommended reference value is the global discard rate (GDR) estimated by Kelleher (2005). Furthermore, the current article proposes the use of two different approaches when using this reference value.

Table 2 Required life cycle inventory items for GDI computation

Items	Requirements	Indicator
Inputs from nature		
Fish catch from fishery	Fish species disaggregation (mass)	GDI_{BRU}/GDI_{mass}
	Mean trophic level per species	GDI_{BRU}
Outputs to the technosphere		
Landed catch	Fish species disaggregation (mass)	GDI_{BRU}/GDI_{mass}
Outputs to nature		
Discards	Fish species disaggregation (mass)	GDI_{BRU}/GDI_{mass}
Additional required information		
• Updated global discard rate from a feasible data source[a]		

GDI global discard index, *BRU* biotic resource use

[a] The specific data source for this research was FAO (Kelleher 2005)

- Biotic GDI method [GDI_{BRU}]. It is based on converting the obtained capture and discard data into BRU values, in order to report final results in terms of removed carbon that was fixed through photosynthesis. GDI_{BRU} involves a characterization phase in which the discarded fish are characterized in terms of NPP, as in a regular BRU calculation procedure, prior to normalization on a world scale.

For GDI_{BRU} calculation, it is necessary to convert the global discard rate and the global landing rates estimated by Kelleher (2005) into net primary production, as shown in Eq. (1) (Pauly and Christensen 1995), where PPR stands for the primary production required and TL for the average trophic level of the selected sample. The selected unit to report PPR calculation was mass of carbon per live weight of fish (g C/kg fish, wet weight).

$$\mathrm{PPR} = [\mathrm{Catch}/9] \times 10^{(\mathrm{TL}-1)} \tag{1}$$

The mean trophic level (MTL) selected for landing rates was set at 3.1, as reported by Pauly et al. (1998a). There are certain limitations when using this number, however, since it may not reflect the current state of world fisheries. Moreover, recent studies suggest that the trophic level of marine webs is underreported through current calculation methods based on catches

(Branch et al. 2010; Caddy 1998). Regarding the mean trophic level for discards, no specific data on a global scale were retrieved from the literature. Furthermore, despite detailed tables in Kelleher (2005) describing the major discarded fish in globally relevant fisheries, no data linked to actual discard breakdown were available. Therefore, the mean trophic level for discards was also assumed to be 3.1 (Pauly et al. 1998a). Nevertheless, any new data that were available in literature may help improve the accuracy and temporal validity of the assumptions when elaborating the reference values for this methodology.

- Catch GDI method [GDI_{mass}]. The catch GDI approach is based on performing GDI calculations in terms of the total amount of catch (e.g., kg of discard/kg of catch), rather than on the energy transfer efficiency from one trophic level to another (Pauly and Christensen 1995). This method implies a simplified and direct method of obtaining a normalized value for discarding without undergoing a characterization stage. While not constituting an orthodox approach within LCA methodological standards, its usefulness may apply when little data related to discard composition are available (i.e., total discard rate for the specific sample).

3.1.4 GDI calculation. Life cycle impact assessment

The final step consists of the environmental characterization of the selected fishery, vessel, or fishing fleet regarding discard rate performance. This stage allows comparison of the environmental impacts of discards for the specific fishery with those given as reference values, and by extension, to any worldwide fishery. The estimation of the GDI, regardless of the selected approach, is conducted through the following equation:

$$GDI = 1/[GDR/((DR \times FU)/LR)] \qquad (2)$$

where GDI is the Global discard index and GDR is the average discard rate for worldwide fisheries. GDR is reported in terms of total mass per selected FU (i.e., grams of C per FU in the biotic GDI approach, or kilograms of discard in the mass perspective). DR is the discard rate (%) for a particular fishery/vessel., FU is the selected functional unit, and LR is the landing rate in a particular fishery/vessel respect to the total capture (%). The selected dimensionless unit created to measure this particular indicator was the global discard unit (gdu).

Regarding the FU entered in the equation, it must be noted that it needs to be converted into the specific mass units that are being used depending on the chosen GDI approach. In other words, if the selected FU is monetary, it would have to be converted to the equivalent values for PPR in the case of biotic GDI or into the equivalent mass weight for GDI_{mass}. Additionally, it is required to insert the GDR value in the same units as the one's selected for the converted FU.

This equation represents the inverse value of the ratio between the worldwide average discard rate and the average discard rate of a given fishery. Its application leads to an index of positive values that can classify fisheries according to three different groups: (a) GDI <1, in which the discard rate for a given fishery is lower than average worldwide discards; (b) GDI=1, in which the discard rate for a selected fishery is equal to that of average worldwide discards; and (c) GDI >1, in which the discard rate for a given fishery is greater than worldwide average values.

3.1.5 Result interpretation

The classification of fisheries based on their discard performance respect to current (or available) world trends is essentially valid for both GDI methods. However, it is important to highlight that while GDI_{mass} presents a linear configuration based directly on the amounts of catch and discard, GDI_{BRU} presents a more complex scenario. In fact, GDI_{mass} will always show a decline in its value for a fishery that is reducing its discards respect to a fixed reference value. In contrast, GDI_{BRU} variations with respect to a fixed reference value may not only reflect discard reductions or increments, but may also show variations in the discard composition. For example, if a fishery shows a decreasing GDI_{mass}, whereas an increasing GDI_{BRU}, more individuals with a higher trophic level are being discarded. This particular example may be observed, for instance, in shrimp trawling fisheries, where the target species (shrimps) will probably have a lower trophic level than many discarded individuals.

While harmonization of the proposed impact category with existing categories appears complex due to the specificity of discards in marine ecosystems, its integration in environmental quality monitoring through damage assessment seems a feasible future perspective. In fact, another approach may entail the construction of a new damage assessment category for marine ecosystems, by integrating a set of fishery-specific impacts. While the latter alternative may imply a skewed analysis of the fishing industry with respect to other industries in life cycle thinking, it may be an attractive option for LCA consideration in fisheries management and eco-labeling schemes (Jolliet et al. 2004).

3.1.6 Recommendations

Discard reporting still lacks transparency and accountability, since discard monitoring is still highly rudimentary. Furthermore, it is costly to improve and to standardize sampling methods worldwide (Lart 2002; Vázquez-Rowe et al. 2011a;

Walsh et al. 2002; Wetherall 2003). Hence, it is expected that future global discard reporting studies may improve the selected reference value and, therefore, the accuracy of the results provided by this index.

An increase in the thoroughness of the discard data inventory, including the detailed breakdown of the discarded catch by species, will benefit the approach that can be implemented for the proposed methodology, as well as the reporting of other fishery-specific impact categories. For instance, the inclusion of discards and other underlying fish consumptions that may occur during fish extraction, such as bait, when analyzing the environmental profile of a particular seafood product would improve the quality of the results for this specific category, since the entire removal of biomass from the ocean for fish extraction would be computed.

3.2 Additional discard indicators

3.2.1 Survival rate of discards

The survival of discarded organisms is essential to understand their potential ecological impact (Kelleher 2005), since it can help understand population dynamics and, when it comes to implementing technological improvements in fishing gears, reduce mortality. Despite there being a broad bibliography on survival rates in fisheries worldwide (Chen and Gordon 1997; Revill et al. 2005), most studies have focused on trawling fisheries, given the strong amount of discards they generate. Furthermore, previous studies suggest that there can be high variance in the mortality of discards, not only from a fishing gear or fishery perspective (Allen et al. 2001; Rodriguez-Cabello et al. 2001), but also between species (Kaiser and Spencer 1995; Revill et al. 2005), as seen in Table 3. For instance, the high survival rate observed for lesser-spotted dogfish (*Scyliorhinus canicula*) with respect to other discarded organisms in trawling hauls is thought to explain the strong proliferation of this species in intensive fishing zones (Walker and Hislop 1998; Revill et al. 2005).

Therefore, the inclusion of an overall survival rate of discards as an endpoint indicator in seafood LCA studies may contribute to contextualize the impact of discards in a specific fishery. High survival rates in a given fishery may suggest reduced concerns regarding the effects of returning the biomass back to sea. In fact, this indicator could be of special interest when eco-labeling a fishery, since some eco-labeling schemes base their assessment on maximum discard rates (Friend of the Sea 2011). Hence, this indicator would help discriminate between "good" and "bad" discards in terms of the reported mortality/survival rates. Nevertheless, the applicability of this indicator, in the same way as other endpoint modeling indexes, is subject to data unlikely to be available for complex multi-species fisheries.

3.2.2 Slipping

Slipping consists of a fishing operation, usually performed in purse seiners, in which part of the catch is freed before drawn aboard (Stratoudakis and Marçalo 2002). The main reason for slipping is linked to size requirements, inadequate characteristics of the individuals, or high grading (Stratoudakis and Marçalo 2002; Borges et al. 2008). From a technical perspective, slipping is not computed as a discard. Moreover, given the lack of a thorough selection on board, it is difficult to quantify its extent in terms of live mass weight, catch composition, and mortality of the released organisms due to net injury or crowding (Huse and Vold 2010).

Nevertheless, small-pelagic species have shown to be strongly affected by gear related injuries, cutting down their chances of survival once slipped (Huse and Vold 2010). Hence, reporting the existence of slipping activities in a specific fishery, as well as the observed or expected[1] mortality/survival rate will assist when evaluating the total biomass that is removed from its natural environment. Reporting of this impact may add valuable information for fishery certification schemes.

3.2.3 Effect of discards on seabird communities

Numerous articles have discussed the effect that discards have on seabird communities (Garthe et al. 1996; Oro and Furness 2002; Furness et al. 1992). Moreover, offal material and/or wastes derived from slipping also influence bird populations. In fact, some reports suggest proliferation of different bird species (scavengers, predators, etc.) depending on the diverging proportions of offal wastes and discards (Furness 2003).

Evaluating the impact due to variation in fish biomass waste has shown to be a complicated procedure, given the difficulty to discriminate between the effects of waste availability in the sea and other ecological processes affecting seabird ecosystems (Votier et al. 2004). Nevertheless, integrated studies evaluating the influence ocean and sea bird ecosystems exert on each other suggest that abrupt cuts in discarding will cause important changes in seabird compositions, without guaranteeing that these shifts will translate into pre-industrial fishing seabird ecosystem structures (Regehr and Montevecchi 1997; Heubeck et al. 1999). While this specific impact linked to discarding and other forms of biomass waste at sea due to fishing are not quantified or evaluated in the present study, it is important to take into account that discard management from a life cycle perspective should consider bird population dynamics as indicators of the health of a particular ecosystem.

[1] The expected limitations to obtain primary data for slipping would probably derive in using bibliographical data for mortality/survival rates.

Table 3 Survival rate range of discards in selected literature publications

Fishery	Species	Survival rate	Reference
Irish Sea (trawling)	Common dab	24%	Kaiser and Spencer (1995)
Irish Sea (trawling)	Plaice	39%	Kaiser and Spencer (1995)
Irish Sea (trawling)	Rays	59%	Kaiser and Spencer (1995)
North Pacific (trawling)	Pacific halibut	26–97%	Kaimmer and Trumble (1998)
Great Barrier Reef (trawling)	Varied fish and cephalopods	2%	Hill and Wassenberg (2000)
Cantabrian Sea (trawling)	Lesser-spotted dogfish	78%	Rodriguez-Cabello et al. (2001)
NE Gulf of Mexico (hook and line)	Atlantic sharpnose shark	90%	Gurshin and Szedlmayer (2004)
Western English Channel (trawling)	Lesser-spotted dogfish	98%	Revill et al. (2005)
New South Wales (trawling)	Southern herring	0–10%	Broadhurst (2008)
New South Wales (gillnet)	Black sole	73–91%	Broadhurst (2008)

4 Application of the proposed GDI indicator to selected fisheries

4.1 Case study: functional unit and system boundaries

A series of examples were proposed based on discard rates reported by recently published scientific papers (Vázquez-Rowe et al. 2010a, 2010b, 2011a, 2011b) to quantify the environmental impact associated with fish landing in Galician fisheries (NW Spain). The FU considered was 1 ton of landed fish in all cases. The FU selection was based on the fact that discards and other fishery-specific impacts are based more on the landings (and catch) of a particular fishery rather than on the landings of a particular species, although it is also true that many fishing gears have varying discard rates depending on the targeted species, season, or area they fish in. Consequently, an FU referred to one specific species would prevent the assessment from getting a realistic perception of the fisheries' performance.

The production system involved exclusively the different operational stages of the fish extraction phase performed by vessels in the selected fisheries (e.g., diesel or ice consumption), as well as the items included in Table 2. Plant materials or onboard post-harvest waste, such as offal, were not included within the discarded material (FAO 1996; Fet et al. 2010). The products were followed from the fishery until landing for sale, constituting a "cradle to gate" analysis (Guinée et al. 2001).

Inventory data were obtained through questionnaires filled out by skippers as reported by Vázquez-Rowe et al. (2011a). Selected case studies included two coastal fishing fleets (trawlers and purse seiners), three offshore fleets extracting at the Northern Stock (trawlers and long liners) and Azores (long liners) fisheries and one trawling fleet working in Mauritanian waters (Table 4).

Data quality for the coastal purse seiners and coastal and offshore trawlers, as well as for the trawling fleet in Mauritania allowed GDI_{BRU} computation, since detailed discard composition breakdowns were available for the mentioned fleets. GDI for the other two fleets were only computed in terms of GDI_{mass}, since their detailed discard composition was unknown. A summary of the LCI for the different fleets is shown in the Electronic Supplementary Material.

4.2 Justification of the case study

The introduction of GDI in fishery LCAs is presented in order to provide a useful methodological innovation to report fishery-specific impacts in seafood assessment studies. In these cases, regular LCA impact categories are not sufficient to provide a complete assessment of the environmental performance of fisheries. Therefore, two additional fishery-specific impact categories were included to broaden the range of environmental assessment, increasing its relevance in fisheries management (Pelletier et al. 2007). Other innovative impact categories, such as prematurely caught organisms (Emanuelsson 2008; Ziegler et al. 2009), were not included due to data limitations.

4.3 Methodology application

Once the LCI stage was complete, the life cycle impact assessment was executed. The software used for computational implementation of the LCIs was SimaPro 7.3 (Goedkoop et al. 2008). The used environmental assessment method was CML baseline 2000. On the one hand, the five commonly employed impact categories in fisheries LCA were included in the assessment (Pelletier et al. 2007): abiotic depletion potential, global warming potential, eutrophication potential and acidification potential, together with the ozone layer depletion potential impact category, which has shown to be more relevant than initially presumed in fishery LCAs due to cooling agent emissions (Winther et al. 2009; Iribarren et al. 2011).

Table 4 Selected Galician fishing fleet samples for the case study

	F1	F2	F3	F4	F5	F6
Sample size	30	24	9	12	9	5
Percentage over total (%)	18.2	23.8	14.3	20.7	33.33	6.4
Year of inventory	2008	2008	2008	2008	2009	2009
Total landings (tons)	12,597	16,056	3,769	3,416	5,000	668
Total captures (tons)	12,998	27,750	6,657	3,473	6,213	727
Target species	European pilchard	Blue whiting	Megrim	European hake	Varied cephalopods	Swordfish
	Horse mackerel	Horse mackerel	Anglerfish	Fork beard	Varied flatfish	Mako shark
	Atlantic mackerel	Atlantic mackerel	European hake	Common ling	Senegal hake	Porbeagle
	Minor species	European hake	Minor species	Atlantic pomfret	Minor species	Bigeye tuna
Reported discards (kg/FU)	32.6	728	766	16.8	243	88.4

F1 coastal purse seining, *F2* coastal trawling, *F3* offshore trawling, *F4* offshore long lining (Northern Stock), *F5* trawling (Mauritania), *F6* offshore long lining Azores

On the other hand, the fishery-specific impact categories and indicators included in the case study were as follows: the two proposed approaches for GDI, biotic resource use, as proposed by Papatryphon et al. (2004), and seafloor impact potential, SIP (Ziegler et al. 2003). SIP development is based on the seafloor impact index proposed by Nilsson and Ziegler (2007). Therefore, the swept seabed area was computed by multiplying the effort by the mentioned index (area swept per hour). As suggested in their study, the calculated area was based exclusively on the area swept by trawl doors and trawl net. BRU and GDI_{BRU} calculation followed the formula provided by Pauly and Christensen (1995). Values for each of the assessed fisheries were based on the trophic levels of the different species that make up the catch composition, including discards, as seen in Table 5 (Pauly et al. 1998a). MTLs of the different species were obtained from *Fishbase* (Froese and Pauly 2011). The MTL for discards with known compositions can be observed in Table 6, while the MTL for discards with unknown composition were assumed to be 3.1 (Pauly et al. 1998a).

4.4 Brief discussion of the case study

Highest values per FU for conventional impact categories corresponded to the offshore fisheries, while the lowest impacts were identified for coastal fleets, mainly the purse seining fishery, which showed the lowest fuel use intensity (Table 7). On the contrary, fishery-specific impact categories did not show this linear correlation. For instance, the SIP for purse seining and long lining fleets was identified as zero, while the values for trawling fleets varied from the coastal trawling fishery (0.68 km^2/FU) to the offshore trawling fleet (4.81 km^2/FU). However, it is important to note that there is a certain correlation between the fishing effort of the trawlers and their potential effect on the seafloor.

The BRU values obtained for the different fisheries were dependent on the catch profile, as well as on the composition of their discards and, when applicable, the use of bait. Therefore, the coastal purse seining fleet that targets small pelagic fish and the Mauritanian trawling fleet that targets cephalopods showed the lowest values since they catch species that are low down in the trophic chain (Pauly et al. 1998a). Moreover, their discards also corresponded to species with low MTL. On the contrary, the offshore long lining fleets were those with the highest BRU values due to the high MTL of the target species, the high amount of bait used per FU and, to a lesser extent due to low discarding in these fleets, discard amounts themselves.

Regarding the GDI_{mass} approach, the highest values (Fig. 3) were identified for the coastal and offshore trawling fisheries (8.86 and 8.36 gdu, respectively), while the best performance regarding discards was that of offshore long liners in the Northern Stock (0.19 gdu). The coastal purse seining fleet also showed a lower discard rate than average (0.38 gdu). Finally, the Azores long lining fleet presented discard values close to the global average (1.02 gdu).

Finally, the obtained results for the GDI_{BRU} methodology were identified for those fleets were discard breakdown composition was available. The value for coastal purse seining was 0.41 gdu, a value very close to that observed in GDI_{mass}, due to the similar MTL of this fleets' catches respect to the values assumed at a worldwide scale. The values for the three evaluated trawling fishing fleets showed a higher value respect to GDI_{mass}, due to the high MTL of the discards respect to the worldwide average (Fig. 4).

Results reflect how conventional impact categories were heavily associated with the energy use in each fishery. This finding is not novel, since it has already been highlighted as the major impact in all the non-artisanal fishing fleets that have undergone LCA analysis (Tyedmers 2001; Hospido

Table 5 Catch composition for the selected fishing fleets and trophic level of the species

Species	Scientific name	TL	Catch (%)
F1=coastal purse seining			
Atlantic mackerel	*Scomber scombrus*	3.65	26.79
Horse mackerel	*Trachurus trachurus*	3.64	23.30
European pilchard	*Sardina pilchardus*	2.61	46.85
Discards	–	3.14	3.16
Total	–		100.00
F2=coastal trawling			
Atlantic mackerel	*Scomber scombrus*	3.65	12.27
Horse mackerel	*Trachurus trachurus*	3.64	10.25
Blue whiting	*Micromesistius poutassou*	4.01	25.10
European hake	*Merluccius merluccius*	4.48	10.24
Discards	–	3.55	42.14
Total	–		100.00
F3=offshore trawling (Northern Stock)			
Anglerfish	*Lophius budegassa*	4.49	17.55
European hake	*Merluccius merluccius*	4.48	8.57
Megrim	*Lepidorhombus spp*	3.69	25.95
Norway lobster	*Nephrops norvegicus*	2.60	0.39
Other species	–	3.00	4.17
Discards	–	3.67	43.38
Total	–		100.00
F4=offshore long lining (Northern stock)			
Atlantic pomfret	*Brama brama*	4.08	16.07
Common ling	*Molva molva*	4.25	9.61
Conger eel	*Conger conger*	4.29	1.79
European hake	*Merluccius merluccius*	4.48	59.68
Fork beard	*Phycis spp.*	3.73	5.75
Rock fish	*Helicolenus spp.*	3.81	5.46
Discards	–	N/A	1.65
Total	–		100.00
F5=trawling (Mauritania)			
Octopus	*Octopus vulgaris*	4.10	63.32
Sepia	*Sepia officinalis*	3.60	9.50
European squid	*Loligo vulgaris*	3.20	9.17
Sole	*Solea solea*	3.13	5.42
Sand sole	*Pegusa lascaris*	3.19	2.53
Senegal hake	*Merluccius senegalensis*	4.50	7.47
Caramote prawn	*Penaeus kerathurus*	2.50	2.59
Discards	–	3.38	19.53
Total	–		100.00
F6=Azores long lining fleet			
Swordfish	*Xiphias gladius*	4.49	31.64
Porbeagle	*Lamna nasus*	4.24	51.24
Blue shark	*Prionace glauca*	4.24	8.25
Bigeye tuna	*Thunnus obesus*	4.49	0.75
Discards	–	N/A	8.12
Total	–		100.00

Table 6 Discard composition for the selected fishing fleets and trophic level of the species

Species	Scientific name	TL	Discards (%)
F1=coastal purse seining[a]			
Atlantic mackerel	*Scomber scombrus*	3.65	27.64
Horse mackerel	*Trachurus trachurus*	3.64	24.04
European	*Sardina pilchardus*	2.61	48.33
Total discards	–	3.14	100.00
F2=coastal trawling[b]			
Atlantic mackerel	*Scomber scombrus*	3.65	7.15
Horse mackerel	*Trachurus trachurus*	3.64	32.54
European hake	*Merluccius merluccius*	4.48	2.18
Blue whiting	*Micromesistimius poutassou*	4.01	8.45
Freckled catshark	*Scyliorhinus spp.*	3.92	8.01
Boarfish	*Capros aper*	3.14	1.46
Haddock	*Melanogrammus aeglefinus*	4.09	1.25
Streaked gurnard	*Trigloporus lastoviza*	3.42	1.97
Invertebrates	–	2.50	15.53
Other[a]	–	3.75	21.46
Total discards	–	3.55	100.00
F3=offshore trawling[c]			
Horse mackerel	*Trachurus trachurus*	3.64	35.00
Pouting	*Trisopterus luscus*	3.73	25.00
Undersized individuals	–	3.99	25.00
Other individuals	–	3.1	15.00
Total discards	–	3.67	100.00
F5=trawling (Mauritania)[c]			
Sardine	*Sardina pilchardus*	2.61	10.00
Cunene horse mackerel	*Trachurus trecae*	3.49	35.00
Chub mackerel	*Scomber japonicus*	3.65	15.00
Other[a]	–	3.38	40.00
Total discards	–	3.38	100.00

TL trophic level

[a] Data corresponding to the purse seining fleet is based on skippers reporting the discard of caught species' juveniles. Therefore, we assumed that these species were discarded in the same proportion as their catch

[b] Discard data for the coastal trawling fleet corresponds to the average discards composition reported by the Galician coastal trawling fleet

[c] Data for offshore trawlers in the Northern Stock and deep-sea trawlers in Mauritanian waters were based on rough estimates elaborated by skippers from the assessed sample

and Tyedmers 2005; Vázquez-Rowe et al. 2011c; Ziegler et al. 2009). However, when discards and the other fishery-specific categories are compared between fisheries this pattern is not as clear, showing that multiple factors can affect a fishery when it is analyzed from an integral perspective. For

Table 7 Characterization values associated with the selected fisheries per FU

Impact categories	F1	F2	F3	F4	F5	F6
Conventional CML 2000 baseline impact categories						
ADP (kg Sb eq)	4.99	12.3	51.4	31.5	41.5	32.2
AP (kg SO_2 eq)	10.1	27.2	115	70.1	93.5	73.8
EP (kg PO_4^{3-} eq)	1.84	4.97	21.0	12.9	17.2	14.2
GWP (kg CO_2 eq)	796	2279	8759	6017	7170	7060
ODP (kg CFC 11 eq)	8.66E-4	7.85E-3	1.72E-2	2.45E-2	1.54E-2	3.88E-2
Fishery-specific impact categories						
GDI_{mass} (gdu)	0.38	8.36	8.86	0.19	2.79	1.02
GDI_{BRU} (gdu)	0.41	23.61	32.74	N/A	5.32	N/A
BRU (g C kg^{-1} fish)	15,838[a]	127,740[a]	121,573[a]	247,681[a,b]	96,784[a]	265,734[a,b]
SIP (km^2)	0	0.68	4.81	0	1.95	0

ADP abiotic depletion potential, *AP* acidification potential, *EP* eutrofication potential, *GWP* global warming potential, GDI_{mass} global discard index, based on live weight catch, GDI_{BRU} global discard index, based on net primary productivity, *BRU* biotic resource use, *SIP* seafloor impact potential, *F1* coastal purse seining, *F2* coastal trawling, *F3* offshore trawling, *F4* offshore long lining (Northern Stock), *F5* trawling (Mauritania), *F6* Azores long lining fleet

[a] BRU results for selected fleets include the BRU of the discards that correspond to 1 t of landed fish

[b] BRU results for F4 and F6 include the NPP relative to the bait used for fishing. However, it does not include potential discards that may have occurred when fishing for the bait

instance, GDI does not necessarily increase with increasing energy use in a particular fishery, BRU depends mainly on the nature of the species being captured and used as bait, and SIP is an impact category that refers mainly to the amount of seafloor dragged per FU by trawlers, while the other gears assessed in this article contribute zero to this impact category.

Previous studies have already highlighted bottom trawling as being responsible for 50% of worldwide discards, while only landing 22% of catches (Kelleher 2005). Nevertheless, this increased discard rate that is common for a great majority of trawling fisheries cannot be linked directly to the increased energy use of these fleets, but must be understood in the specific context of each fishery. Hence, despite the fact that an elevated energy use usually implies long dragging hours that may increase discards, compared to shorter gear operations (e.g., purse seining), there are other, more significant aspects that may influence high discard rates, such as the gear used itself, the targeted species, the overexploitation of the fishery or a variety of hydrographical factors.

Additionally, it is important to highlight the pyramidal structure of marine ecosystems. Therefore, the NPP that generates in the lower level of the trophic chain tends to move upwards, with a consequent loss of a high percentage of the productivity due to growth or spawning of marine organisms (Villasante 2009). The use of GDI_{BRU}, which is recommended whenever quality data is available, allows the inclusion of discard reporting in terms of net productivity appropriation from the sea through fishing activities that is not destined to human/industrial consumption, but is returned, usually with an excessively high mortality rate, to the sea (Cappell 2001).

Hence, the proposed GDI category for this case study was implemented in order to include a feasible and universal discard indicator for fishing fleets. Moreover, it provides a dynamic value easily comparable between fisheries, facilitating the understanding of discard evolution through time and the proposal of specific discard mitigation management policies (Levasseur et al. 2010). The implementation of this index, therefore, will be useful to assess the effects of the increased use of the catch, evaluate if there are any improvements in the selectivity of the fishing gear, and detect changes in catch composition over time.

4.5 Recommendations and advantages of the methodology

Currently, fishery LCAs still present an important number of unresolved challenges. This study attempts to provide contributions to partly resolve methodological gaps. In particular, GDI is presented as a midpoint index to provide an additional criterion for eco-efficiency based on discard quantification. It also guarantees worldwide applicability, even though discards, in the same way as other fishery-specific issues, do not describe fisheries homogeneously at a world scale (Byrd et al. 2011; Hall et al. 2000; Johnsen and Eliasen 2011). However, the fact that it constitutes a midpoint indicator may reduce its relevance in terms of decision support (Bare et al. 2000).

Another relevant characteristic of GDI is the fact that it is a flexible index. Since its calculation involves direct comparison with average discards worldwide, a global reduction of discards will translate into a worse GDI value for a particular fishery, provided that the discard rate for this fishery does not

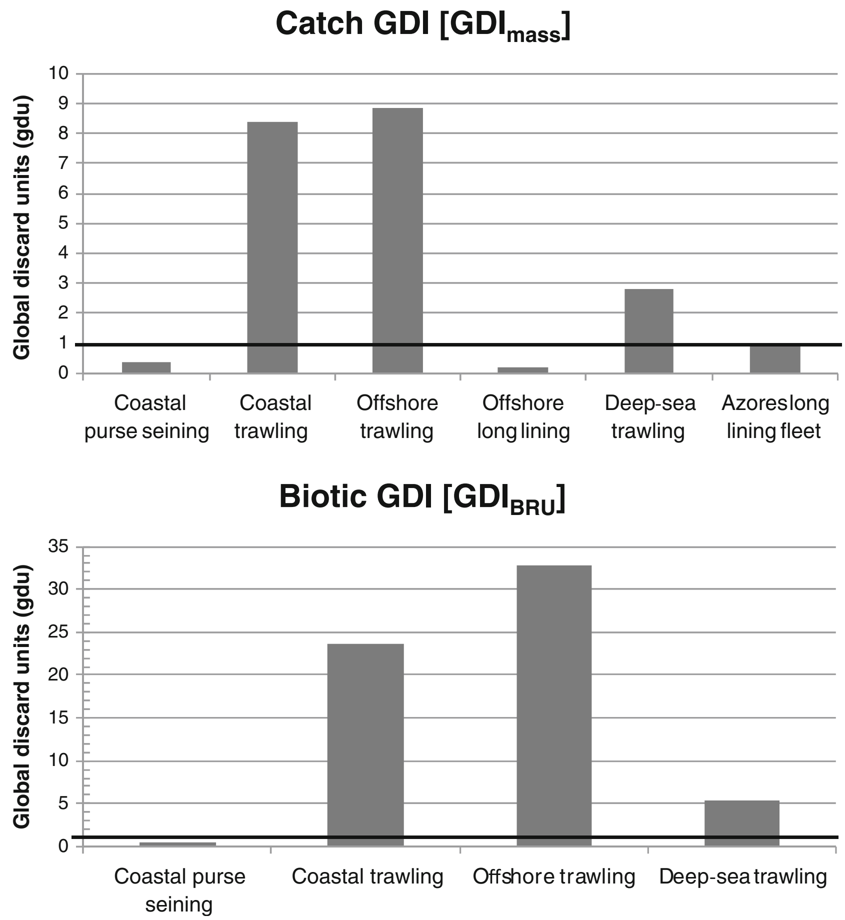

Fig. 4 GDI results for the selected fishing fleets. *Black horizontal line* indicates GDI= 1, which represents worldwide GDI mean

decrease and assuming, if GDI_{BRU} is applied, that there are no variations in the MTL for discarded biota. This advantage may translate into an important starting point for fisheries management when it comes to analyzing how discard reduction policies applied elsewhere may be used in a particular fishery.

The inclusion of GDI as a regular indicator in fishery LCAs may also enhance discard reporting in this type of assessments. This implementation would not only increase the availability of results relating to discards in worldwide fisheries, but will also increase the reporting of discards in fisheries with suspected low discards. This is an important point, since low discarding fisheries are not necessarily linked to lower impacts on the ecosystem (Kelleher 2005; Zhou et al. 2010). Furthermore, it also provides increased feasibility when reporting discards, avoiding this concept to be used interchangeably or as an equivalent term to by-catch, as occurs in many scientific publications (Kelleher 2005), making it difficult to determine whether the values presented refer to landings or to total capture.

Finally, discard reporting has shown to be highly variable on a temporal scale, despite an unquestionable discard reduction trend through time in many fisheries (Catchpole et al. 2011; Kelleher 2005; Vázquez-Rowe et al. 2011a). Therefore, an ideal analysis should be based on adequate chronological series. However, currently most fishery LCA case studies, including the one presented in this article, fail to analyze discards and other fishery aspects, such as stock assessment or fishing effort, on a prolonged temporal scale, due to difficulty in achieving wide inventories for a great number of years (Ramos et al. 2011).

4.6 Limitations of the methodology

The main limitations linked to the implementation of GDI_{BRU} are associated with the calculation of the average MTL for individual species, as well as at a global scale. Despite considering a total of 220 different species groups of invertebrates and fish, based on FAO landing statistics, MTL values do not discriminate between increasing MTL of individuals as they age. This characteristic of marine species has not been addressed in depth in the available literature, so MTL values refer to average trophic levels per species (Caddy 1998; Pauly et al. 1998b; Pauly and Palomares 2005). Moreover, values reported by Pauly and Christensen (1995) do not account for illegal, unregulated, and unreported fishing. Therefore, it is expected, e.g., that in fisheries with important discard amounts regulated with thorough minimum size requirements, the MTL of the discards may be overestimated due to the lower value

expected in the younger, undersized individuals that are being discarded.

5 Conclusions

The methodology presented in this study achieved the important objective of integrating discard data as a fishery-specific impact in LCA studies. The application of GDI, as well as the other two suggested indicators, may not succeed at determining the specific impacts that discards are potentially causing in a particular ecosystem, since discard patterns depend on a wide range of variables (stock abundance and distribution, fishery policy, fishermen behavior, etc). Nevertheless, they constitute a useful starting point to identify tendencies in discard and discard management in worldwide fisheries, thanks to their dynamic characteristics.

In fact, to date, conventional impact categories used in LCA have focused on the environmental impacts that originate from the wide range of fishery-linked industrial activities. However, the driving force of most of these is linked to energy use. Therefore, the proposed methodology, similarly to other biological categories (i.e., BRU or SIP) aims to increase the usefulness of fisheries LCA as a management tool, by adding a fishery-specific perspective to the assessment. Accordingly, the specific indicators can be understood as a regular procedure to follow in fishery assessment.

Nevertheless, future research will have to determine how midpoint indicators proposed for fishing systems are integrated consistently for damage assessment in LCA. One possible approach may consider the construction of a new damage category for fishery-specific impacts, which may trigger the usefulness of LCA in fisheries management. An opposing perspective, however, would consider their integration in currently existing damage categories, in order to avoid comparability gaps in LCA interpretation.

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